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Implementing ecological potential of lakes for the Water Framework Directive—Approach in Flanders (northern Belgium)

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ABSTRACT

The European Water Framework Directive stipulates that artificial and heavily modified water bodies must reach good ecological potential, as opposed to good ecological status as required for natural water bodies, if certain hydrological and morphological pressures cannot be removed. We propose to obtain these objectives from the conditions expected once all feasible mitigating measures have been taken but relative to the internationally agreed quality goals for natural lakes. This allows a more objective, consistent and transparent definition of the possible deviation from the ultimate quality level than prevalent methods, in particular those considering the improvement expected from future measures. A level of functional integrity should be endeavored that minimizes the probability of undesired phenomena and negative consequences for biodiversity objectives embodied by the Habitats Directive. This remains a difficult exercise, but as a first approximation, constraints for the secondary alteration of lake hydrochemistry and the consequent potential for development of submerged vegetation can be considered. Six case studies illustrate possible procedures, which may be further refined to include additional relations between hydromorphological pressures, physical–chemical conditions and biota as knowledge develops or circumstances require.

Introduction

Management of all water resources in the European Union is regulated by the Water Framework Directive (WFD; EC, 2000). Its application to surface waters hinges on assessment of their ecological status for several biological quality elements (BQEs) and of chemical status for a number of specified pollutants. This must be based on a reference system where the highest category (high ecological status, HES) includes water bodies whose ecology, chemistry, hydrology and morphology are not noticeably affected by human activities. Other water bodies (WBs) must be restored to good ecological status (GES), defined as only slightly different from high status. Normal ecological functioning should also remain safeguarded (CIS, 2003), implying unhindered migration of natural biota and absence of nuisance phenomena or health hazards. To measure distance to target, the relative deviation from the biological reference condition needs to be considered, mostly in terms of altered species composition and abundance or biomass of key biota, also accounting for an increased probability of undesired phenomena for aquatic life and amenities.

Defining ecological status is difficult (*e.g.* Furse et al., 2006; Solheim et al., 2008; Solimini and Sandin, 2012) and many data have been collected in the attempt (see Hering et al., 2010). Ecological quality is not necessarily accurately reflected by simple measures of community composition and there are difficulties in determining the characteristics of high status when most available sites have been severely changed. Artificial and heavily modified water bodies (both further on referred to as HMWBs) add further complications. They must meet 'good ecological potential' (GEP), whilst maintaining the functions for which they were created or modified (*i.e.* navigation, recreation, water storage and regulation, flood protection, land drainage or other equally important sustainable development activities). Similar to GES, this objective may differ slightly from the best possible condition, maximum ecological potential (MEP). The Directive gives little guidance as to the meaning of these 'potentials' but indicates that they will be water-body specific and as close as possible to the most comparable natural situation. Only modifications in hydromorphology (HM; a conjunction for hydrological and morphological conditions) for which elimination would be technically unfeasible, or have adverse consequences for the specified uses or the wider environment, may be tolerated and mitigation of negative effects should be maximal (EC, 2000; CIS, 2003; Borja and Elliot, 2007).

Member states are obliged to (re)set pertinent long-term objectives by 2015 for all WBs, heavily modified or not, in their River

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Basin Management Plans, even though the necessary knowledge to underpin such decisions will often be limited. The WFD specifies that adequate protection should be given to sensitive WBs and requirements of other directives, such as for the Habitats Directive (HD), must also be met (EC, 1992; ECOSTAT, 2005; Mainstone et al., 2008; Irvine, 2009). Revised objectives for HMWBs should not result in failure to obtain favorable conservation status of habitats or species listed by the HD (Uttley, 2010; Schmedtje and Kremer, 2011). As ecological potential usually involves a lower quality level than ecological status and c. 25% of all European water bodies are heavily modified or artificial (DG, 2006), with a considerably higher proportion in some countries (Kampa and Laaser, 2009), the risk of not meeting this provision is substantial.

With this in mind, we illustrate practical procedures to set ecological goals for heavily modified and artificial lowland lakes with six examples from Flanders (northern Belgium). These represent three different situations with regard to conservation requirements, each with its own possible allowances to ecological potential: 1/lakes within a N2000 Special Area of Conservation (SAC) and designated as important for a protected eutrophication-sensitive aquatic habitat type, 2/not within a SAC but presenting such a habitat, and 3/neither of the former.

Typically, modifications of morphology, water level and flow are related to aquatic and riparian habitat suitability when assessing HM pressures on lakes (Ostendorp et al., 2004; Hellsten and Dudley, 2006), but overall hydrochemistry may also be affected. As this is common in lowland lakes, particular attention is given to ecological potential where this appears to be the case.

Methods

Conceptual framework

Two lines of reasoning are officially endorsed to define GEP (Fig. 1). The first one starts from MEP by estimating the effects from HM pressures that will remain once all appropriate (= technically feasible and without adverse effects) mitigating measures have been taken. This can be based on modeling, comparison or expert judgment (CIS, 2003). It requires three steps for each BQE: (1) identification of the MEP for HM considering designated uses and other provisions, (2) estimation of the resulting physical–chemical MEP in relation to the most comparable water type, and (3) estimation of the ensuing biological MEP. In this way, MEP values are set on the ecological quality scales as HES values minus the HM effect sizes. Next, the biological GEP is identified as a slight deviation from MEP and its supporting physical–chemical values are established.

Aiming to avoid possibly unrealistic objectives and the compounding of errors from estimating both reference conditions and the distance between HES and MEP, an alternative approach followed (Kampa and Kranz, 2005; Irmer and Pollard, 2006). This so-called pragmatic method considers the ecological conditions prevailing in the HMWB and estimates the improvement that can be obtained for each BQE if two sets of measures are applied: (1) all appropriate mitigation measures, and (2) only such measures deemed significantly rewarding, exempting all those which together would yield only slight improvement. Addition of these estimated gains to the actual status results in MEP and GEP, respectively. Thus, GEP becomes independent of MEP but is determined by the effort considered worthwhile. Allegedly, the level of ambition would be similar to the CIS method, but clearly this would depend on a very scrutinous and uniform selection of measures that can be confidently ignored, the resources allocated to mitigation and the extent to which this is allowed to interfere with existing uses – all this within as well as between member states.

In view of the lack of experience with both approaches, DG (2006) calls for further demonstration of their advantages and disadvantages and recommends further work. An overview of their use reveals both are equally popular (Kampa and Laaser, 2009) but Hering et al. (2010) note that there is no consensus which method should prevail.

In the approach adopted here, ecological potential objectives are also determined from the HM pressures that remain when all mitigating measures have been applied and not on preselected measures (Fig. 1). However, both ecological potential levels are derived directly from the corresponding type-generic goals for BQEs, using the same assumptions or models to estimate effect sizes. Thus, MEP is defined relative to HES, and GEP relative to GES. By using the internationally calibrated benchmarks for natural WBs, GEP and its distance from MEP are defined on a more consistent and, hopefully, implicitly meaningful basis. Additionally, GEP is constrained by certain requirements to safeguard biodiversity objectives and public health. We also demonstrate that MEP and GEP can often be established without actually modifying the expected values for ecological quality ratios (EQRs) used for status classification by adjusting assessment practice where pressure effects are confounded and poorly predictable.

Overall, HMWB objectives will depend on a set of criteria. These include: (1) direct effects of HM pressures, *e.g.* from spatial restrictions or water-level fluctuations limiting the occurrence of a BQE, (2) indirect effects through altered hydrochemical conditions, (3) concordance of GEP with goals for the HD (or additional national legislation, if pertinent), and (4) constraints by risk of undesirable disturbances. The following examples illustrate the modalities such as they have implemented in relation to the Flemish BQE assessments (VMM, 2009) in more detail.

Study systems

Six lakes with a surface area of 50–85 ha and average depth up to 11 m in the central eastern part of Flanders were examined (Fig. 2; Tables 1 and 2). Although modest in dimensions, these are among the larger standing WBs in the region. Lake Vinne (VI) is the only natural one among them. It was restored in 2005 from a completely drained basin that served as a poplar plantation for more than a century. All five others are artificial. Lakes Kessenich (KE) and Heerenlaak (HE) are gravel pits, whereas Grote Vijver (GV) and Schulensmeer (SM) originate from sand extraction. Consequently, their depth profiles mostly show steep gradients. The saucer-shaped Lake Blokkersdijk (BD) results from the incomplete raising of a polder with Tertiary deposits dredged from the River Schelde in the early 1970s to allow industrial development. All these WBs are about 40 years old. According to the WFD typology for Flanders (Denys, 2009), KE, HE and GV are comparable to stratified alkaline lakes with a moderately high natural nutrient status (Aw-e), whereas VI, BD and SM are closest to permanently mixed alkaline lakes with a moderate mineral content and equally fairly eutrophic background (Ami-e). The pressures from modified hydrology, morphology and recreation acting upon the lakes are listed in Table 3. Inputs from heavily modified rivers, water-level fluctuations and various structural amenities prevail. Lakes KE, HE, GV, SM experience boating and angling; BD and VI are important water-bird sites with restricted access.

Sampling and analysis

Surface area and shore length were determined from aerial photographs by GIS analysis, other morphological variables from in situ measurements and detailed bathymetric maps (Lowrance LMS-522c iGPS sonar and natural neighbor interpolation). Residence

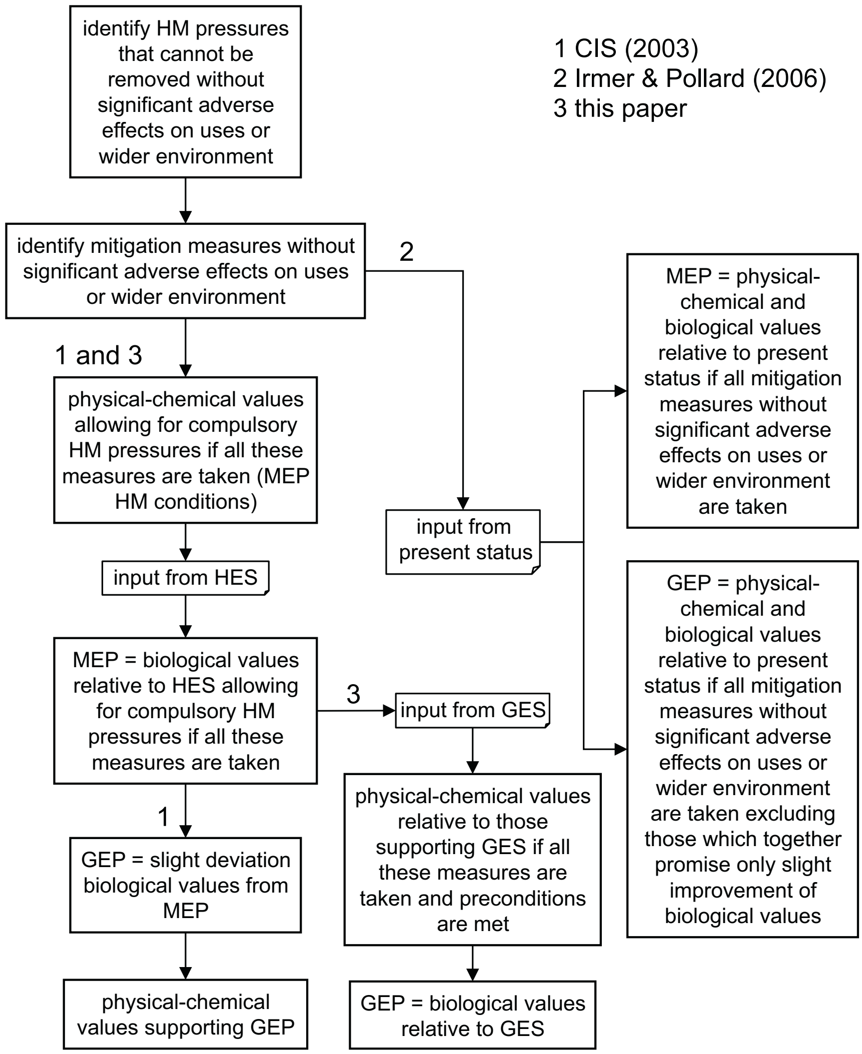

Fig. 1. Procedures to define maximum (MEP) and good ecological potential (GEP) for lake water bodies, using high (HES) and good ecological status (GES) or present condition as reference levels.

time was calculated from volume, estimated ambient precipitation, evaporation, river inflows, outflows and groundwater influx.

Data on physical–chemical conditions and phytoplankton were collected on a monthly basis in April–October 2007 (KE and HE) and 2008 (BD, GV, VI, SM). At each occasion, Secchi and thermocline depth were determined from a boat by Secchi-disk and multiparameter probe (YSI 600 QS-08) measurements at 50 cm intervals. The entire epilimnion was sampled at 16 random locations in each lake with tube and bottle samplers and equal volumes from each location (5 L) were combined to a single sample. Conductivity and pH were measured at the site (YSI 600 QS-08). Alkalinity (titrimetric according to Golterman and Clymo, 1969), suspended particular matter (gravimetrically after filtration through a Whatmann GF/F filter), chl a (2–4 h acetone extraction after point sonication and analysis according to Van Heukelem and Thomas, 2001), major ions (Dionex ICS 2000 ion chromatograph), total phosphorus, soluble reactive phosphorus, total nitrogen and ammonium–nitrogen (Skalar Analytical autoanalyser) were determined within 24 h. Subsamples of 250 mL were fixed following Sherr and Sherr (1993) for analysis of the phytoplankton composition using inverted microscopy according to the CEN (2006) standard. At least 400 cells or coenobia were enumerated and identified to the lowest taxonomic level possible. Floating colonies of cyanobacteria were assessed from Sedgewick-Rafter counts. Picocyanobacteria were quantified by epifluorescence microscopy according to MacIsaac and Stockner (1993). Phytoplankton densities were converted to biovolumes and carbon biomass, using factors given by Menden-Deuer and Lessard (2000). Calculation of the EQR from average chl a and relative biomass of cyanobacteria followed Van Wichelen et al. (2005).

Samples for diatom periphyton were taken on a single day in September 2007 (KE, HE) or July 2008 (VI, GV, BD, SM) from different places along the shore, usually by collecting at least 10 reed stems (*Phragmites australis* (Cav.) Trin. ex Steud.) that faced the open water on each location and scraping the part that was c. 20 cm below the water surface. After treatment with hydrogen peroxide and hydrochloric acid, permanent slides were prepared, using Naphrax. Exactly 500 randomly selected valves were identified from individual samples to species or lower taxonomic level to estimate relative abundance of impact-associated and impact-sensitive diatoms. At least three spatially separated samples were analyzed and additional samples counted until the standard deviation on the average phytobenthos EQR was $\leq$0.2 units. Lake EQRs were calculated as average site scores (Hendrickx and Denys, 2005).

Macrophyte composition and abundance were surveyed in July–August along the entire exposed shore and for the entire surface of each WB up to 4 (stratified lakes) or 2 m (shallow lakes) below the water surface. These depth limits correspond to the expected minimum colonization depth for natural lakes in Flanders derived from estimated minimum transparency within the

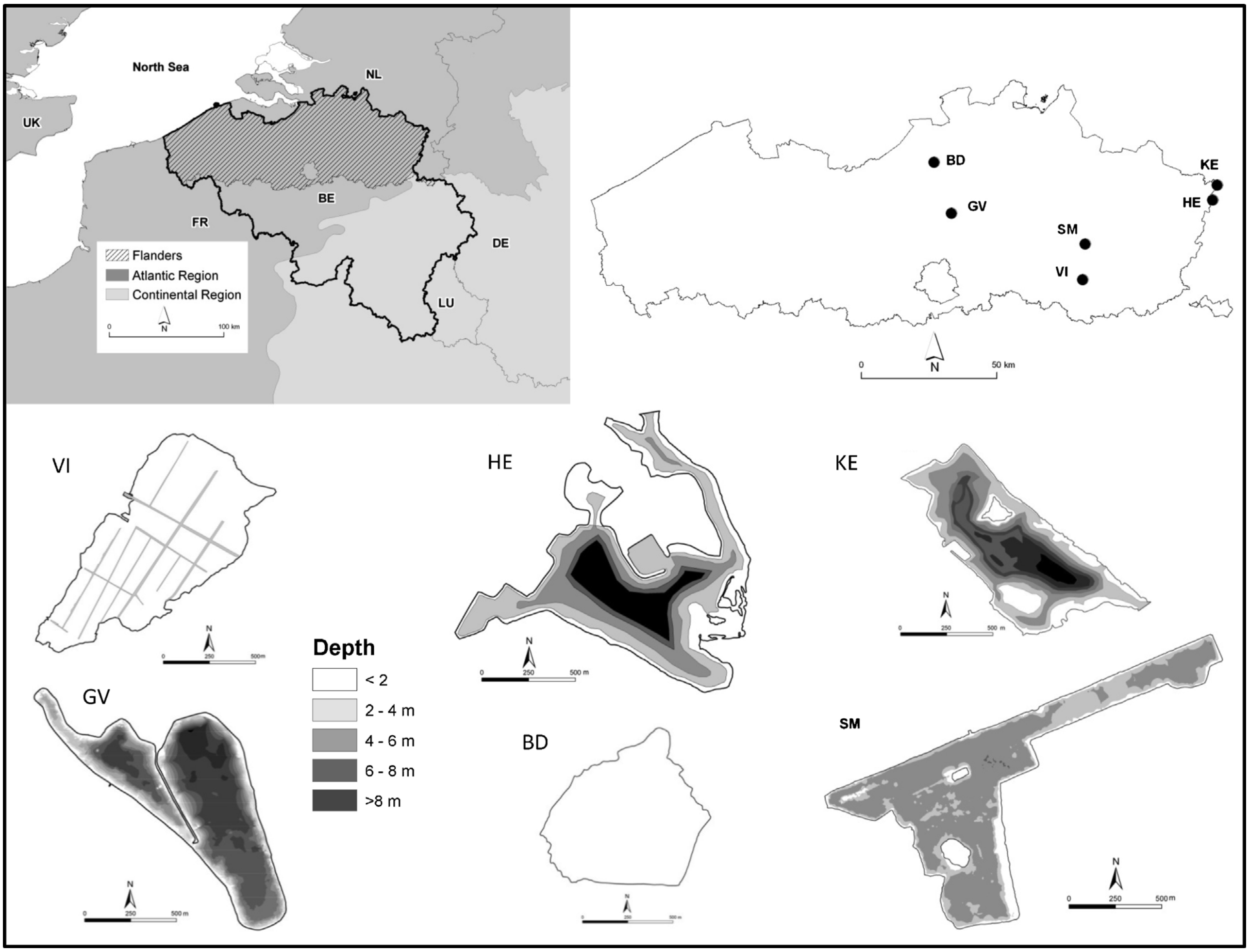

Fig. 2. Situation of lower Belgium (Flanders region) in Europe with location of the studied lakes and their bathymetry (VI Vinne, HE Heerenlaak, KE Kessenich, GV Grote Vijver, BD Blokkersdijk, SM Schulensmeer).

unimpacted nutrient range. Separate relevés were made for parts with homogeneous vegetation of aquatic or riparian vegetation using a 5-point abundance scale. Total abundance of submerged plants in each part was assessed on the ECOFRAME scale (Moss et al., 2003). Macrophyte EQRs include metrics for type-specific species composition, representation of disturbance-indicating taxa, growth-form diversity and abundance of submerged vegetation (Schneiders et al., 2004; Leyssen et al., 2005).

Benthic macroinvertebrates were sampled semi-quantitatively in July using a 50 cm wide hand net with 0.5 mm mesh. All major mesohabitats (open water, submerged macrophytes, helophytes and soft substrate) were sampled along the shore at five different locations. Complementary collections were made by hand from hard substrates. Samples were preserved in 95% ethanol with 5% ether, and processed separately in the laboratory. EQR assessment followed Gabriels et al. (2010) with metrics for total number of taxa, number of Ephemeroptera, Plecoptera and Trichoptera taxa (EPT), number of other sensitive taxa, Shannon diversity index and average tolerance. The average score of all site samples served as the lake EQR.

Estimation of nutrient concentration, chlorophyll a and macrophyte colonization depth

Where considered appropriate, total phosphorus concentrations for MEP and GEP were estimated using Vollenweider-type mass balance models, either assuming no phosphorus sequestration to the sediment (Lake BD) or a loss rate that fitted best to a large set of European and North American lakes (Brett and Benjamin, 2007: 9, hypothesis 4; Lake SM). Resulting chl a was derived from empirical relations. Where control by phosphorus was expected (GEP for Lake SM), a limiting ratio for the 95th percentile of Dutch lakes without dominance of cyanobacteria was used (Portielje and van der Molen, 1999, p. 379, Table 3). In case of phytoplankton suppression by additional factors, such as zooplankton (MEP and GEP for Lake BD, MEP for Lake SM), an average relation for Danish lakes was applied (Jeppesen et al., 1997, p. 154, Table 3). Assuming that an increase of suspended inorganic solids and detritus above the type-specific average at GES should be averted by mitigation, the corresponding Secchi depth was estimated (Roozen et al., 2003, p. 522, Eq. (3)). Maximum colonization depth of rooting submerged macrophytes was predicted according to Middelboe and Markager (1997, p. 558, Eq. (5)). Depending on the relevant vegetation composition, parameters for charophytes (Lake BD) or caulescent angiosperms (Lake SM) were used.

Results

Lakes with aquatic Natura 2000 habitats in a SAC

SACs serve as biodiversity strongholds and hubs for reestablishment of threatened species and constitute focal areas for active habitat improvement. Lake restoration usually displays hysteresis, and nutrient levels below pre-disturbance conditions are required

Table 1
Environmental characteristics of the studied lakes.

Lake	Vinne (VI)	Heerenlaak (HE)	Kessenich (KE)	Grote Vijver (GV)	Blokkersdijk (BD)	Schulensmeer (SM)
Origin	Natural; restored after long period of complete drainage	Artificial; gravel extraction	Artificial; gravel extraction	Artificial; sand extraction	Artificial; raised polder	Artificial; sand extraction
Matching water type	Ami-e	Aw-e	Aw-e	Aw-e	Ami-e	Ami-e
Inflows	2	1	1	–	–	1
Outflows	1			1	–	1
Sampling year (April-October)	2008	2007	2007	2008	2008	2008
Surface area (ha)	69.9	84.9	52.8	62.2	47.9	76.8
Volume (m^3)	0.8×10^6	4×10^6	6×10^6	6.2×10^6	0.4×10^6	3×10^6
Average depth (m)	1.2	4.7	11.3	10	0.74	4
Maximum depth (m)	2.5	14.9	26	18.3	1.4	6.4
Shore length (km)	5.3	10.4	4.8	5.7	3.1	7.2
Residence time (days)	279	26	89	4818	252	212
pH[a]	7.8 (6.5–8.5)	8.2 (6.5–8.5)	8.6 (6.5–8.5)	8.5 (6.5–8.5)	9.8 (6.5–8.5)	8 (6.5–8.5)
Conductivity (μS/cm)[a]	611	395	420	717	473	387
Alkalinity (meq/l)[a]	4.8	2.2	2.5	1.9	3	1.9
Secchi depth (cm)[a]	61 (90)	117 (180)	133 (180)	522 (180)	Bottom	126 (90)
Suspended particular matter (mg/l)[a]	17.4	8.8	9.1	3.7	17.4	8.3
Thermocline depth (m)[a]	–	6.6	6.9	6.6	–	–
TP (μg/l)[a]	118 (70)	109 (55)	100 (55)	55 (55)	824 (70)	81 (70)
PO_4-P (μg/l)[a]	12	22	21	2	682	3
TN (mg/l)[a]	5.7 (1.3)	3.6 (1.3)	4.6 (1.3)	1.4 (1.3)	3.2 (1.3)	2.7 (1.3)
NH_4-N (μg/l)[a]	474	102	63	61	341	50
SO_4-S (mg/l)[a]	12.1 (33.3)	10.1 (33.3)	10.9 (33.3)	59.1 (33.3)	12.3 (33.3)	45.8 (33.3)
Chlorophyll a (μg/l)[a]	26.7	8.7	14.5	6.9	36.6	40.6
Charophytes	×	–	–	×	×	–
N-2000 habitat present	HT 3150	HT 3150	HT 3150	HT 3140	No	No

Aw-e: alkaline stratified with moderately high nutrient background.
Ami-e: alkaline mixed with moderate mineral content and moderately high nutrient background.
[a] Mean of monthly measurements during the sampling period; physical–chemical values supporting GES of most comparable water type between brackets.

to suppress turbid conditions and allow macrophyte recovery. Consequently, there are compelling reasons to apply more stringent nutrient targets for SACs than elsewhere to sustain the full range of regional habitat conditions, support less-tolerant associated biota and maximize restoration success. The nutrient conditions supporting HES are therefore considered most appropriate as default long-term objectives for SACs (Van Looy et al., 2008).

An example is Lake VI, situated in a SAC designated important for the Natura 2000 habitat 'Magnopotamion and Hydrocharition of naturally eutrophic lakes' (Fig. 3). The enriched organic topsoil and most poplar trees were left in place prior to refilling of the lake. This represents an important alteration of substrate conditions but does not warrant a change in objectives as it can still be remediated, given appropriate finances. Likewise, the long-lasting absence of permanent water needs no consideration. The only permanent HM modifications are an artificial limitation of the highest water levels to prevent flooding of a farm house classified as national heritage and loss of some riparian area to necessary installations. The latter affects macrophytes in a small part and this is excluded from their assessment. The slight restriction of water-level variation does not compromise any of the BQEs.

Table 4 gives an overview of objectives and assessment results. For phytoplankton, an average chl a concentration of 11 $\mu g\,L^{-1}$ or less is required but current values are more than double, resulting in a qualification as poor. High nutrient levels (average TP 118 $\mu g\,L^{-1}$, TN 5.7 $mg\,L^{-1}$) are to blame. The phytobenthos, nevertheless, suggests a condition corresponding to good or better. Limited development of rooting submerged plants results in moderate macrophyte conditions. This is caused by low transparency (average Secchi 0.6 m) and difficult establishment on organic sludge and debris. Intensive decomposition also leads to rather poor oxygen conditions and moderate status for macroinvertebrates. Along with measures to reduce external nutrient inputs, removal of the organic top soil and wood debris is considered most rewarding to improve conditions. Measures to reduce guanotrophication by a breeding colony of black-headed gulls, a declining species in Europe, are more controversial.

Lakes with aquatic Natura 2000 habitats not within a SAC

Where no specific goals have been designated but habitat still needs to be conserved, the general physical–chemical conditions of GES, including nutrients, are used to support a favorable local conservation status. This condition applies to three of our sites (Fig. 3).

Lakes HE and KE are both permanently connected by a single channel to the River Meuse. Both have the same habitat type as Lake VI, but no special conservation ruling applies. Sizeable water-level fluctuations occur, but their duration during the vegetation season remains too limited to have a notable effect on any of the vegetation metrics. Based on literature and regional observations, a negative effect would be expected in Flemish lakes deeper than 2 m if their amplitude exceeded 2 m for more than 6 days within a month of the vegetation season (May to September). In this case, only sparse submerged vegetation on gravelly substrates and a somewhat lower growth-form diversity with absence of more sensitive nymphaeids (van Geest et al., 2005; Paillisson and Marion, 2006) are taken for granted, if not refuted by on-site observations. Macrophytes would need to be excluded if prolonged water-level changes exceeded 4 m – the expected minimum colonization depth for stratified lake types. In this case, the BQE would be represented solely by phytobenthos collected from floating samplers. Until their relation to water-level changes is known better, objectives remain unchanged for macroinvertebrate metrics. Consequently, both lakes are essentially assessed as natural, except that parts where macrophyte growth is compromised by recreation (HE) or temporary gravel storage (KE) are not considered for this BQE.

Table 2
Numerically most abundant taxa in the studied lakes (PC picocyanobacteria).

Lake	Vinne (VI)	Kessenich (KE)	Heerenlaak (HE)	Grote Vijver (GV)	Blokkersdijk (BD)	Schulensmeer (SM)
Phytoplankton	*Chrysococcus minutus*, *C. rufescens*, *Cryptomonas erosa*, *Didymocystis inconspicua*, PC, *Syncrypta eleochrus*	*Aulacoseira granulata*, *Chroomonas acuta*, *Microcystis aeruginosa*, *Pseudoanabaena mucicola*, PC, *Stephanodiscus hantzschii*	*Chroomonas acuta*, *Cyclotella*, *Cryptomonas*, PC, *Stephanodiscus hantzschii*	*Chlorella vulgaris* f. *minuscula*, *Chroomonas acuta*, *Chrysochromulina parva*, PC	*Aphanizomenon flos-aquae*, *Chroomonas acuta*, PC	*Cryptomonas erosa*, *Geitlerinema* sp., *Limnothrix redekei*, PC, *Stephanodiscus hantzschii*
Periphyton diatoms	*Achnanthidium minutissimum*, *Gomphonema parvulum*, *Encyonopsis microcephala*	*Cocconeis placentula* var. *euglypta*, *C. placentula* var. *lineata*, *Navicula cryptotenella*, *Rhoicosphenia abbreviata*	*Eolimna minima*, *Cocconeis placentula* var. *lineata*, *Rhoicosphenia abbreviata*	*Achnanthidium minutissimum*, *Cocconeis placentula*, *C. placentula* var. *lineata*, *Gomphonema micropumilum*, *G. minusculum*	*Cocconeis placentula* var. *lineata*, *Gomphonema parvulum*, *Nitzschia paleacea*	*Achnanthidium minutissimum*, *Cocconeis placentula* var. *lineata*, *Cyclostephanos dubius*, *Gomphonema parvulum*, *Nitzschia paleacea*, *N. subacicularis*
Macrophytes	*Lemna minuta*, *Phragmites australis*, *Potamogeton berchtoldii*, *Riccia fluitans*, *Utricularia australis*	*Elodea nuttallii*, filamentous algae, *Potamogeton pectinatus*	*Potamogeton pectinatus*	*Chara contraria*, *Elodea nuttallii*, *Potamogeton pectinatus*, *P. pusillus*	*Chara globularis*, *Enteromorpha* sp., *Phragmites australis*, *Potamogeton pectinatus*, *Zannichellia palustris*	*Ludwigia grandiflora*, *Phragmites australis*, *Najas marina*
Cladocera	*Daphnia galeata*, *D. pulex*, *Simocephalus vetulus*	*Bosmina coregoni*, *Daphnia cucullata*, *D. galeata*, *Diaphanosoma brachyurum*	*Bosmina coregoni*, *Daphnia galeata*, *D. hyalina*	*Daphnia galeata*, *D. hyalina*, *Diaphanosoma brachyurum*	*Daphnia magna*, *D. pulex*, *D. pulicaria*	*Bosmina coregoni*, *Daphnia galeata*, *Diaphanosoma brachyurum*
Macroinvertebrates	Chironomidae, *Cloeon*, *Physella*, *Sigara*	Chironomidae, *Bithynia*, *Valvata*	Chironomidae, *Micronecta*, *Pisidium*, *Sigara*, *Valvata*	Asellidae, *Dreisena*, Gammaridae, *Potamopyrgus*	*Cloeon*, Hydroptilidae, *Micronecta*, Naididae, *Plea*, Tubificidae	Chironomidae, *Cloeon*, *Dreissena*, *Limnomysis benedeni*
Fish	*Gasterosteus aculeatus*, *Pseudorasbora parva*, *Pungitius pungitius*	*Anguilla anguilla*, *Gymnocephalus cernuus*, *Rhodeus amarus*, *Rutilus rutilus*	*Anguilla anguilla*, *Blicca bjoerkna*, *Rhodeus amarus*, *Sander lucioperca*	*Anguilla anguilla*, *Blicca bjoerkna*, *Perca fluviatilis*	*Cyprinus carpio*, *Gasterosteus aculeatus*, *Scardinius erythrophthalmus*	*Anguilla anguilla*, *Rutilus rutilus*

Nutrient concentrations are above GES/GEP objectives in KE and HE (Table 1). The condition of KE is judged moderate for phytoplankton and phytobenthos, both indicating eutrophication, and good for macroinvertebrates, but only poor for macrophytes (Table 4). The latter is due to dominance of the invasive *Elodea nuttallii* (Planch.) St. John (Table 2). Obvious solutions to this are lacking, although improved nutrient conditions may offer some relief (Josefsson, 2011). Limiting inflow of nutrient-rich river water to peak discharges would increase relative groundwater influence and help to attain GEP for phytoplankton and phytobenthos. In HE, phytoplankton and macroinvertebrates reach a level comparable to GES, whereas phytobenthos and macrophytes classify as moderate (Table 4). Note that excluding as much as 17.5% of the area from aquatic vegetation assessment raises the EQR by only 0.07. Next to the water quality of the Meuse, the main reasons for an insufficient ecological potential are some local agricultural drainage, but also reduced transparency from turbulence in shallower areas. Yachting necessitates a navigable connection to the river, so mitigating measures are limited to the reduction of near-shore scouring by creating protective structures and more gentle slopes, as well as firmer supervision of boating regulations.

The water level of Lake GV is not influenced by the tidal regime of the adjacent River Dijle and remains quite stable. However, the lake will become a basin for controlled water storage. River

Table 3
Hydromorphological pressures on the investigated lakes.

Lake	Vinne (VI)	Kessenich (KE)	Heerenlaak (HE)	Grote Vijver (GV)	Blokkersdijk (BD)	Schulensmeer (SM)
River inputs	–	×	×	×	–	×
Amplified water-level variation	–	×	×	–	–	×
Restricted water-level variation	×	–	–	–	–	–
Impervious shore-line protection	–	–	×	–	–	×
Weirs, boating facilities	–	–	×	×	–	×
Angling facilities	–	–	–	–	–	×
Artificial catchment soil	–	–	–	–	×	–
Gravel storage	–	×	–	–	–	–
Vegetation management	×	–	–	–	–	–
Water skying	–	–	×	×	–	–
Motor boating	–	–	×	×	–	–
Angling	–	–	×	–	–	×

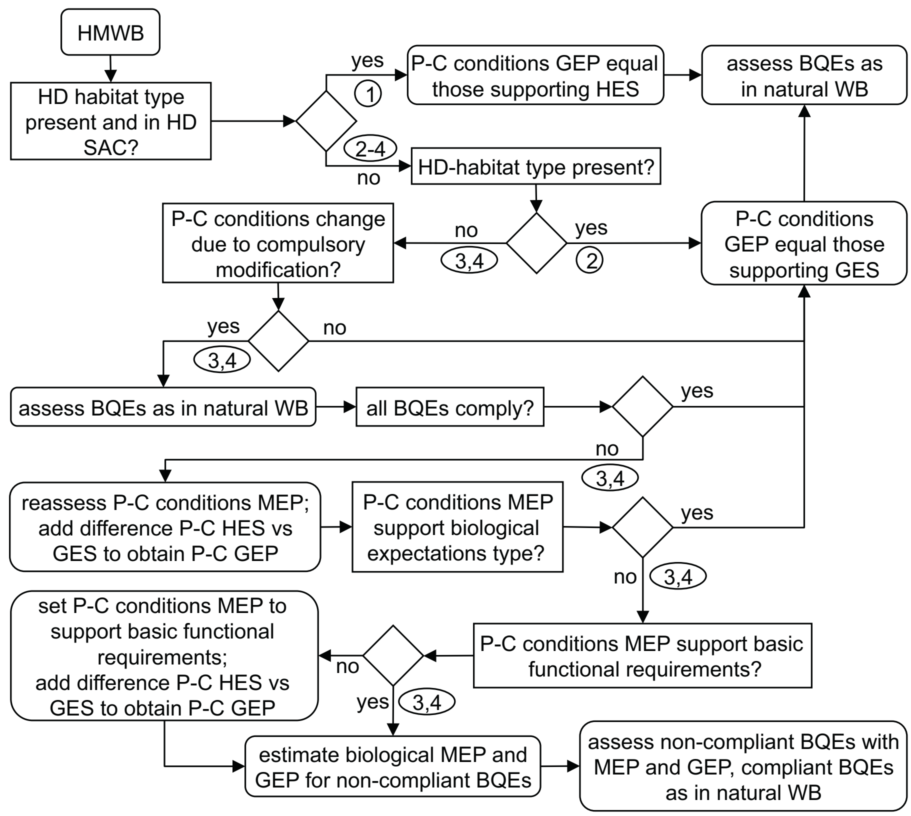

Fig. 3. Procedure to derive maximum (MEP) and good ecological potential (GEP) where a change in hydromorphology affects physical–chemical (P–C) conditions. Numbers indicate pathways for case studies discussed in the text: (1) Vinne, (2) Heerenlaak, Kessenich, Grote Vijver, (3) Blokkersdijk, (4) Schulensmeer.

water is expected to enter once every 50 years, or so, after passing an extensive wetland that should retain most suspended matter and nutrients. With such low frequency and compliant river-water quality, these events are unlikely to impose a marked pressure. As with both previous lakes, presence of a protected habitat type, 'benthic *Chara* vegetations of oligo-mesotrophic waters', sets nutrient goals at those for GES and expectations for BQEs need no reassessment in this respect. Extensive bank stabilization with boulders, turbulence from water skiing and docking facilities for boats are the main HM alterations/pressures from designated uses.

Current TP and TN are borderline for GEP (Table 1) and EQRs for phytoplankton, phytobenthos and macrophytes are all within the range of GES (Table 4). For macroinvertebrates, the potential remains moderate due to low scores for EPT and Shannon diversity index at sites with uniform habitat and heavy surges. Possibilities to develop more natural shoreline features and reduce wave action exist for most part of the lake and can alleviate this condition. Nutrient conditions require vigilance and will benefit from better sealing of the outflow culvert which allows some polluted river water to enter at high tide. Moreover, general quality would benefit from excluding motor boats from the western part of the lake which presents higher turbidity and sensitivity to disturbance.

Lakes without Natura 2000 habitat objectives

Objectives are allowed to change more extensively for sites BD an SM, where no additional protection measures are required (Fig. 3).

Although a Ramsar site and within a special protection area of the Birds Directive, Lake BD is not part of the HD network, nor does it present any of its specified habitat types. The HM change that needs consideration here is an artificial catchment soil of displaced Tertiary marine sands and silts. These deposits release very high amounts of phosphorus. Because the phosphorus content is not a relic of man-made pollution, this represents a secondary pressure from catchment alteration that may alter ecological potential if it affects the BQEs. Presence of industrial plants prevents replacement of the soil by a less problematic substrate. No WB-specific objectives are set for artificial ground WBs, so the general quality target for groundwater imposed by decree of the Flemish Government (BVR, 2010) of 1.34 mg L^{-1} PO_4 applies. Without legal grounds for improving ground-water quality beyond this level, this value is assumed to be the ultimate average P load of groundwater delivered to BD and is used to estimate conditions at MEP (Fig. 3). Considering that little P is lost to the lake sediments due to wind-stressed conditions and intense bird activity, as supported by observations, an in-lake TP concentration of no less than c. 140 $\mu g\,L^{-1}$ is expected at MEP. For GEP an additional 30 $\mu g\,L^{-1}$ would be allowed, corresponding to the difference in TP levels supporting HES and GES. The estimated corresponding average amounts of chl a are 66 $\mu g\,L^{-1}$ at MEP and 78 $\mu g\,L^{-1}$ for GEP. Charophytes, mainly *Chara globularis* Thuill., are an important part of the present vegetation and their light requirements are used to assess the expected colonization depth of submerged plants: respectively 1.4 m at MEP and 1.2 m at GEP. Hilt et al. (2006) suggest that at least 50% cover may be necessary to stabilize clear-water conditions in shallow lakes and we consider this as a cautionary value that can be used to set MEP where more reliable lake-specific estimates are unavailable. Obviously, the corresponding lake area varies with morphometry, but for very shallow lakes with a maximum depth up to 2 m, such as BD, a provision of 50% potential cover at MEP can be implemented throughout. Importantly, preconditions on potential submerged vegetation occurrence also provide a link to the level of phytoplankton development and nutrient conditions. Due to the shallowness of the basin, the chl a estimate at MEP of 66 $\mu g\,L^{-1}$ is not expected to limit macrophyte growth in BD to less than 50% of its surface

Table 4
EQR scores and status classification for BQEs of the lakes according to their ecological potential as HMWB or for natural conditions (only one EQR is shown in case of identical assessment). Classification: H, high; G, good (both for natural lakes only), G+, good or above (for HMWBs only); M, moderate; P, poor; B, bad.

Lake	Assessment	Vinne (VI)		Kessenich (KE)		Heerenlaak (HE)		Grote Vijver (GV)		Blokkersdijk (BD)		Schulensmeer (SM)	
		HMWB	Natural	HMWB	Natural	HMWB	Natural	HMWB	Natural	HMWB	Natural	HMWB	Natural
Phytoplankton	Chl a	0.34 (P)		0.48 (M)		0.64 (G+/G)		0.72 (G+/G)		0.96 (G+)	0.45 (M)	0.66 (G+)	0.42 (M)
Phytoplankton	Cyanobacteria	0.93 (G+/H)		0.54 (M)		1.00 (G+/H)		0.78 (G+/G)		0.29 (P)		0.41 (M)	
Phytoplankton	Overall	0.34 (P)		0.48 (M)		0.64 (G+/G)		0.72 (G+/G)		0.29 (P)	0.29 (P)	0.41 (M)	0.41 (M)
Phytobenthos		0.67 (G+/G)		0.59 (M)		0.46 (M)		0.78 (G+/G)		0.25 (P)	0.23 (P)	0.44 (M)	0.44 (M)
Macrophytes	Shore	0.84 (G+)	0.84 (H)	0.86 (G+)	0.86 (H)	0.84 (G+)	0.84 (H)	0.83 (G+)	0.83 (H)	0.78 (G+)	0.78 (G)	0.92 (G+)	0.92 (H)
Macrophytes	Aquatic	0.43 (M)	0.43 (M)	0.35 (P)	0.34 (P)	0.55 (M)	0.48 (M)	0.63 (G+)	0.63 (G)	0.60 (G+)	0.35 (P)	0.13 (B)	0.09 (B)
Macrophytes	Overall	0.43 (M)	0.43 (M)	0.34 (P)	0.34 (P)	0.55 (M)	0.48 (M)	0.63 (G+)	0.63 (G)	0.60 (G+)	0.35 (P)	0.13 (B)	0.09 (B)
Macroinvertebrates		0.67 (M)		0.83 (G+/G)		0.77 (G+/G)		0.59 (M)		0.70 (G+)	0.70 (G)	0.61 (M)	0.61 (M)
Overall		0.34 (P)	0.34 (P)	0.35 (P)	0.35 (P)	0.47 (M)	0.47 (M)	0.59 (M)	0.59 (M)	0.25 (P)	0.23 (P)	0.13 (B)	0.09 (B)

(≈0.65 m) and is consequently considered acceptable; otherwise, the phytoplankton objective would have needed readjustment.

Actual TP values are five times higher than those supporting GEP, implying that ecological potential cannot exceed moderate (Table 1; CIS, 2005). Nevertheless, *Daphnia* grazing (Table 2) keeps average chl a at 37 μg L^{-1}, well below the required level. Dominance of cyanobacteria in the late-summer phytoplankton, however, leads to a poor status (Table 4). Unlike chl a, cyanobacteria metrics are not adapted in ecological potential. For phytobenthos, the expected EQRs at MEP and GEP are based on the 75th percentiles of observed abundance values for indicative diatoms in WBs with TP concentrations below 140 or 170 μg L^{-1}, respectively. Even with this relaxation, the EQR corresponds to a classification as poor. Submerged macrophytes extend below the depth expected for MEP and need no specific treatment in this respect. Moreover, two other prominent species, *Potamogeton pectinatus* L. and *Zannichellia palustris* L. (Table 2), belong to the expected vegetation of such hypereutrophic conditions. As a result, they are not considered to influence classification negatively as in comparable lakes with unmodified catchment. No similar exception is made for lemnids and *Enteromorpha*, however, because their proliferation would suppress rooting submerged plants and challenge appropriate lake functioning. Given these exceptions, the macrophyte community is judged to attain a marginally good ecological potential, at least in the survey year. For macroinvertebrates, an EQR of 0.7 represents good status, corroborating that no allowances need to be made here. Clearly the main concern for BD is nutrients. Leaching will eventually decrease P inputs, but in the meantime measures for reducing P loading warrant consideration. Immobilization of P in the catchment soil or construction of a permeable reactive barrier may be the only possible ones. Even so, it may very well prove difficult to attain the proposed GEP in the more proximate future, as migratory birds also contribute substantially to the massive nutrient concentrations. If more favorable conditions develop, persistence of adequate vegetation will depend on the level of phytoplankton suppression that can be maintained which might be assisted by food-web manipulation.

Lake SM lies within a SAC, but is not designated for the conservation of eutrophication-sensitive habitats and such habitat is also absent. The main HM pressures result from its water-storage function at peak discharges of the River Demer. This induces water-level fluctuations of up to +3 m but these last too briefly to affect macrophyte occurrence significantly. Small parts of the banks are not available for colonization by plants and left unconsidered. More importantly, the physical–chemical objectives of this reservoir need to be aligned to those of the water it receives (Fig. 3). Contrary to Lake BD where only one reference level is at play, both MEP and GEP for the tributary inflow determine the lake objectives. With respective TP values at c. 90 and 210 μg L^{-1} for the river, the average lake concentrations for SM are estimated to increase with c. 16 and 38 μg L^{-1} above the type-specific standards, yielding 56 μg L^{-1} TP at MEP and 108 μg L^{-1} at GEP. These differences are too small to necessitate a reassessment of the phytobenthos or macroinvertebrate expectations. If phytoplankton is phosphorus-limited and suspended matter does not increase above usual at GEP, the TP values translate to c. 19 and 82 μg L^{-1} chl a. The lake is not stratified but substantially deeper than 2 m, with the shallower area only representing 12% of its total area. This means the present bathymetry would not allow an 'acceptably safe' submerged cover without demands on transparency that are substantially higher than requested at GES for the most comparable water type. The expected colonization depths for submerged caulescent macrophytes at the calculated chlorophyll concentrations are 2.9 m for MEP, but only 1.4 m for GEP. This means the type-specific expectation for a minimum colonization depth of 2 m can hold for MEP, but for GEP this objective is lowered to 1.4 m.

Although the average current TP of c. $80\,\mu g\,L^{-1}$ appears acceptable already, autumnal values are considerably higher. Phytoplankton is considered moderate due to development of cyanobacteria in late summer (Table 2) when toxic surface blooms recurrently prevent recreation (Descy et al., 2011). Results for phytobenthos and macroinvertebrates are similar. However, macrophytes are the most critical BQE in the classification of Lake SM. A very limited development of submerged plants up to only 1 m depth, mainly the turbidity-tolerant *Najas marina* L., corresponds to no more than a bad ecological potential (Table 4). Measures to improve ecological potential will first need to concern the water quality of the River Demer, which causes severe episodic pollution and nutrient build-up in the lake sediment. Within the lake, facilitation of macrophyte development is essential and expansion of shallow areas up to c. 1.4 m will be a priority measure to this end. Clearly, the risk for cyanobacterial blooms is still considerable at the nutrient concentrations supporting GEP (Carvalho et al., 2011) and additional measures, *e.g.* timely flushing (Verspagen et al., 2006; Paerl et al., 2011), may be necessary in their event.

Discussion

In contrast to rivers, transitional waters and coastal areas (DG, 2006; Duursema et al., 2006; Borja and Elliot, 2007; Pottgiesser et al., 2009; Breine et al., 2011), ecological potential for lakes has received little attention. So far, only Lammens et al. (2008) sketched the use of the pragmatic approach for some Dutch lakes.

Borja and Elliot (2007) pointed out that GEP will be prone to substantial variation depending on factors that have no ecological basis. We also consider this a major point of concern that will compromise international comparability of efforts. GEP intends to support the least detrimental continuation of activities that are valued more by society than restoration of ideal nature values. Given the leverage of social drivers, socioeconomic considerations will usually be more decisive than environmental issues in determining an acceptably 'slight deviation' from MEP as required in the CIS approach, as well as in the selection of measures yielding only 'slight improvement' for its pragmatic alternative. The latter method appears to be particularly sensitive in this respect, offering better prospects to evade the cumbersome process of obtaining a derogation (CIS, 2009). Ideally, the same measures should be implemented in both approaches recommended by ECOSTAT, but one can assume that without prior commitment to certain ecological targets, water managers will only consider the least expensive or conflicting ones, if any at all (Keessen et al., 2010), when confronted with opposing interests. Usually, selection of measures proceeds from pick lists (Kampa and Laaser, 2009) and techniques not yet embedded in administrative procedures will be less likely to receive attention. Measure efficiency itself also varies strongly among lakes (*e.g.* Penning et al., 2012) and is difficult to predict in the absence of validated lake-specific models. Long-term results of restoration efforts suggest an overly optimistic appreciation (Gulati et al., 2008). Consequently, the choice of sufficiently rewarding measures against those that in combination offer little improvement is likely to vary between managing authorities and member states. Although here also, the difference between MEP and GEP should only be 'slight', it is not clear how this can be consistently confined. As pointed out by Pollard and Huxham (1998) and Moss (2008a), terms such 'slight deviation' or 'slight improvement' are ambiguous and bound to lead to different interpretations. GEP and MEP are biological conditions, the difference between which can be conceived in many ways. Even from the perspective of values on an EQR scale, ambiguity remains considerable, if not just because only the high/good and good/moderate boundaries are leveled by international calibration and GEP is situated below this quality range. Moreover, EQRs from different member states often mirror very different biological sensitivities (Birk et al., 2012) and confounding of HM effects with other pressures in evaluating current status can influence estimation of possible improvements. Thus, considerations on current condition will also add to international disparities. In short, using a selection of measures to identify goals within continuously changing socioeconomic constraints facilitates a sliding base line and renders a comparable level of ambition unlikely. Although the CIS (2003) method draws profit from the internationally agreed benchmark of HES, it is similarly burdened with the consideration of a slight degree of difference to define GEP.

Our alternative approach extends on the original CIS procedure by explicitly considering the difference between the physical–chemical values that support the intercalibrated biological HES and GES of a particular water type to ensure more transparent, stable and comparable goals. In addition, thresholds for key characteristics are used to avoid excessive concessions on biodiversity and ecological quality or public health. The non-linear translation of differences in physical–chemical values at MEP and GEP to the gap between biological MEP and GEP values allows the 'slight' divergence of MEP and GEP to vary in line with biological response and methodological sensitivities. As with ecological status, estimates of ecological potential rely on perceived natural references. For some man-made systems with rather unique characteristics and a short history of continuous change, such as the IJsselmeer discussed by Lammens et al. (2008), this may be rather elusive. In such rare cases, an equally unique set of semi-natural reference conditions – a water type with a single WB – might be established from comparable HMWBs close to MEP (CIS, 2003) or expert judgment, if necessary (Martin et al., 2011). Such objectives may be open to revision in successive River Basin Management Plans (CIS, 2003, 2009).

A second point of concern is that, even though ecological potential does not include aspects of pollution, there remains a threat that physical–chemical changes resulting from HM changes cause conflict with other environmental goals, public health provisos, or amenities for sectors of less immediate economic importance. Mechanisms are desirable to minimize this possibility. At the moment, there are only few guidelines to set environmental requirements in order to sustain functional integrity and avoid nuisance phenomena, *e.g.* development of nuisance algae, toxic cyanobacteria, anoxia, etc. Water-quality standards intended to serve legislation for general pollution abatement, such as the Nitrates and Urban Wastewater Treatment Directives (EC, 1991a,b), serve a different purpose. Increased abundance of cyanobacteria and appearance of toxic strains have important consequences for wildlife, public health and many water uses. Their emergence is poorly predictable (Kardinaal and Visser, 2005; Søndergaard et al., 2011) but not acceptable in view of existing recommendations (WHO, 2003, 2004) and national regulations. Therefore, GEP for cyanobacterial biomass was always kept at the value of the type-specific GES in phytoplankton classification. More generally, turbid conditions due to phytoplankton development reduce lake amenities and are widely accepted as undesired for most lakes. Fixed thresholds for mean summer chl a might therefore be considered to confine MEP. However, dissimilar conditions occur at similar concentrations and very high temporal variability makes this end-of-chain criterion difficult to apply (Søndergaard et al., 2011). Chances are also that conditions have already deteriorated beyond autonomous return once this point has been reached.

An option is to consider more essential structural and functional attributes of reference conditions to improve ecosystem resilience, as suggested by Article 2 of the WFD and emphasized by, *e.g.*, Moss (2007), Kelly et al. (2009) and Irvine (2011). For many lakes, the

propensity to support a certain abundance of submerged plants reflects the functional capacity to sustain an acceptable status (Jeppesen et al., 1998; Scheffer, 1998; Burks et al., 2006) and, as far as not restrained by designated uses, this should be reflected in MEP. We propose that lake conditions at MEP should allow the cover of submerged vegetation to be maximized as much as possible. Considering the currently vegetation-deprived state of many lentic WBs in Flanders, this might be seen as stringent or even unrealistic. Yet only lakes with very high natural turbidity or presenting limiting physical impairments beyond mitigation should be truly exempt if the concept of ecological potential is assumed to stimulate any environmental improvement and enhance resilience to additional pressures, *e.g.* such as resulting from climate change (Mooij et al., 2007; Nõges et al., 2007; Moss et al., 2011). Positive feedbacks of submerged macrophytes on the transparency of shallow lakes in regions with a moderate climate already start at fairly low cover (*e.g.* Meijer et al., 1999; Janse et al., 2008), but given the associated uncertainties and substantial between-lake variability (Søndergaard et al., 2010), a precautionary high proportion of potentially covered area – *e.g.* 50% – can be an appropriate default for the MEP of very shallow lakes where morphology poses no serious constraints. For lakes of somewhat greater depth and where this requirement does not impose higher demands than for natural lakes, we consider that transparency at MEP should allow colonization by submerged vegetation to a depth that enables 50% submerged cover. Alternatively, lake-specific estimates of the preconditions for vegetation abundance that provide adequate functional resilience could be made where possible. We acknowledge that even so the step from potential cover to realized cover may be large and will depend on many additional factors (Bakker et al., 2012). Similar prerequisites on coverage seem less essential for stratified lakes and would often necessitate major morphological alteration. Although the importance of submerged macrophytes in such lakes for providing habitat complexity and resources to other biota is also well established (*e.g.* Bry, 1996; Schmieder et al., 2006) and a relation to clear water conditions should also not be dismissed (Hilt et al., 2009), it is even less evident to suggest any 'safe' level of macrophyte abundance than for shallow lakes. In Flanders, macrophyte colonization in natural stratified lakes is expected to a depth of at least 4 m. Considering that development is often sparse in the shallowest most turbulent or helophyte-dominated parts, we provisionally accept that rooting submerged plants warrant a basic level of habitat integrity if they occur up to c. 2 m depth. Hence, absence of light limitation is required up to this depth for MEP, confining possible nutrient increases from HM alteration for these WBs.

In our case studies, the focus was primarily on phosphorus limitation and not on other possible changes in chemistry that may arise from hydromorphological pressures. Because nitrogen levels that would put type-specific ecological goals at risk – as far as not resulting from pollution – are mitigated more effectively than phosphorus, *e.g.* through catchment measures, wetland construction and lake management (Jeppesen et al., 2011) they are not included in our estimation of ecological potential. Likewise, any increase of mineral turbidity from HM changes is assumed to be mitigated down to natural levels in all instances dealt with so far. In some cases, effects of raised mineral turbidity may, however, need to be accounted for.

Given budgetary restrictions, it may take considerable time to deploy all the necessary measures that are not considered disproportionately costly (CIS, 2009) and often a stepwise approach will be taken. Biology in particular may respond slowly. For key criteria in regular assessment, such as cyanobacteria dominance, this should not remain veiled by a GEP that allows sub-standard environmental conditions. Instead proper documentation and explanation of shortcomings should be given to argue temporary exemption as outlined by CIS (2009) and additional effect-oriented measures considered.

The WFD and the HD have their own finality (Irvine, 2009), but it is necessary to integrate both for efficient water policy (Mainstone, 2008). As such, a more stringent delineation of GEP may be required for WBs with eutrophication-sensitive habitats and its objectives should be included in appropriate assessment according to Article 6.3 of the HD. We suggest that GEP should not deviate from GES for aspects of water quality where Natura 2000 habitats are concerned, and that even more ambitious baseline objectives (HES) can be applied within SACs. This concurs with the provision of better protection for sensitive WBs, and prioritizing high quality areas for conservation and – most importantly – successful restoration, which requires lower pressure thresholds than mere persistence.

Assessments presented in this paper are based on single lake year observations and, similar to any long-term objectives, are prone to uncertainty. Several of the lakes we studied experience marked year-to-year variations that reflect in EQR values. Most HM pressures also show considerable and often highly irregular temporal variation (*e.g.* KE, SM). Especially BQEs presenting higher turnover can vary consequently and will probably require more intensive sampling than the minimal frequency of 3 years required by the WFD for non-compliant WBs. Moreover, choice of appropriate BQEs for operational monitoring needs to take their sensitivity with regard to HM pressures and mitigation measures into account. In spite of sometimes considerable modification in HM, we usually obtained overall EQR values close to those for an assessment of natural conditions (Table 4). Lake assessments remained similar because permanent HM pressures were subordinate to a precarious water quality. Also, objectives were redefined rather conservatively, the more so where effect sizes need further corroboration. As demonstrated by HE, even sizeable loss of potential macrophyte habitat does not necessarily imply much slacker restoration demands for this BQE. Scores for aquatic macrophytes only improved markedly in case of an extreme secondary effect on nutrient status (viz. BD).

The procedures outlined in this paper, by necessity relying on mechanistic oversimplification, require further elaboration. Undoubtedly, considerable improvements can follow as water managers are confronted with various field situations and data from WFD monitoring become available. For most BQEs and EQRs the relations to HM conditions remain obscure and only crude estimations can be made at the moment. More reliable ways of determining the biological consequences of HM modification and combined pressures can be expected from topical investigations and integrative ecosystem modeling (Mooij et al., 2010). For some well-studied lakes, system-specific models may already provide higher accuracy. Even though 27% of all assessment methods developed so far is reported as sensitive toward hydromorphology, this mainly concerns rivers and transitional waters (Birk et al., 2012). Methods deployed for lakes primarily aim at detecting changes in trophic status and it is difficult to foresee how expectations for such EQRs will be affected by the concept of ecological potential, more so as pressure-impact relationships need much closer examination. Given the limited opportunities for international intercalibration (Pollard, 2011), a transparent methodology and adherence to some common principles for setting quality objectives therefore become even more crucial. Moss (2008b) considers the concept of an ecological potential well intended, but unlikely to yield much real improvement for hydropower reservoirs that are 'disabled' in connectivity and in structure by the lack of a true littoral. This fortunately does not necessarily apply to many situations in the European lowlands. Where more degrees of freedom are left open, sound definition of ecological potential should still allow important quality gains.

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